






Risk Assessment and Decision Making on Mitigation Measures

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15.1 Introduction

Sustainable development aims for decarbonized renewable energy to combat the impacts of climate change. Hydropower is one mode of renewable energy that actually contributes about 16% to the global gross electricity production (IEA 2020). Capturing energy from rivers is in line with the revised EU Renewable Energy Directive (Directive (EU) 2018/2001 of 11 December 2018), which established at least 32% share of

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the Union's gross final consumption as new binding target for the EU by 2030, with a possible upwards revision already in 2023.

Directive 2018/2001/EU also advocates measures to support small renewable schemes through direct price initiatives such as feed-in tariffs (Article 17). Such incentives supported refurbishment and new installation of renewable energy schemes, including small-scale hydropower schemes in EU Member States. However, hydroelectricity generation causes a variety of environmental effects (compare Chap. 4) causing trade-offs between carbon emission-free energy generation and environmental impacts, which contradict other EU policies like the Water Framework Directive or the Biodiversity Strategy. Still, there is no commonly agreed, standardized and reproducible environmental risk and impact assessment of hydroelectricity generation to inform decisions on commissioning new or refurbished hydropower plants.

This chapter presents the first comparable environmental hazard scoring tool for risks and impacts of hydropower plants on fishes and further guidance for assessing cumulative effects of consecutive barriers and hydropower plants within a river to support informed decisions to mitigate environmental impacts from hydropower.

15.2 The European Fish Hazard Index

The trade-off between renewable hydroelectricity and environmental risks and impacts of hydropower plants (HPPs) on river ecosystems needs careful consideration at every single hydropower scheme. A suitable assessment framework evaluates hazards of one hydropower constellation relative to others as a function of their specific operational, constructional and technical characteristics and the ambient fish community, its sensitivity and species-specific mortality risk.

The European Fish Hazard Index (EFHI, van Treeck et al. 2021), is a fish-based assessment tool for screening the risks of hydropower for fishes and meets these requirements. It translates conceptual and empirical knowledge about hydropower-related hazards for fishes into site-specific risk scores. The EFHI is designed as modular assessment framework that offsets the relative hazard of a planned or existing HPP with the susceptibility of the local fish assemblage. It explicitly considers specific autecological characteristics of local species (van Treeck et al. 2020), their conservation value and specific regional management targets. The EFHI is applicable across all European biogeographic regions and can be adjusted to local environmental conditions and conservation objectives by selecting particular target species. At the same time, the EFHI is comprehensive and sufficiently versatile to be applied to a wide range of HPP designs in various stream types.

The EFHI integrates both, species-specific sensitivities of the fish community derived from species' life-history traits and conservation value as well as specific operational, constructional, and technical characteristics of an HPP. Principally the EFHI's hazard

components and the final EFHI score increase with increasing severity of operational, constructional, and technical hazards of the HPP and with increasing sensitivity of the affected species community. The highest possible EFHI scores will be assigned to hydropower plants posing the highest overall mortality risks to fishes, installed in streams with numerous sensitive or conservation-critical species.

EFHI considers five constructional, technical and operational aspects of hydropower schemes that directly affect fishes: (i) upstream and downstream flow alterations, (ii) entrainment risk, (iii) turbine mortality, (iv) upstream fish passage, and (v) downstream fish passage (Fig. 15.1). These factors and their specific hazards are extensively described by van Treeck et al. (2021). Therefore, this chapter only briefly refers to the EFHI's mechanistic functioning. The EFHI considers hazards in relation to the characteristics of the ambient, site-specific fish community. This is captured by using the sensitivity of species to additional mortality (van Treeck et al. 2020) to weigh hazards.

The EFHI requires information about the: (i) HPP's main dimensions, turbine specifications, operating conditions, fish migration facilities and fish protection installed, (ii) target species, and (iii) river reach characteristics. Hazard thresholds were derived from conceptual and empirical knowledge or model results and subsequently categorized to

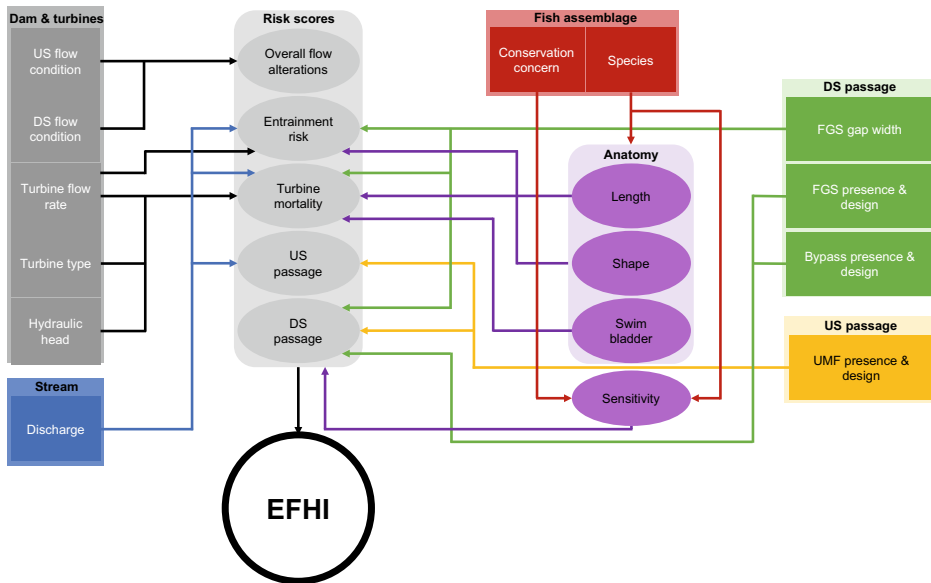


Fig. 15.1 Mechanistic model of the European Fish Hazard Index components. Rectangles = input parameters, ellipses = derived variables, open circle = final index. The “risk scores” box represents the process of transferring risk classes, species sensitivities and anatomies into adjusted and unadjusted hazard- and species-specific risk scores as shown in Table 15.1, which are aggregated to the EFHI. Figure obtained from van Treeck et al. (2021)

“high”, “moderate” or “low” risk. These categories were cross-tabulated and weighed by the rounded integer value of the species’ biological sensitivity (obtained from van Treeck et al. 2020) and yield a numerical score for each hazard and target species as shown in Table 15.1. Component- and species-specific hazard scores can take values from 0 to 1 in 0.25 steps, with higher scores indicating more severe hazards. Up to five target species can be selected to best reflect the local fish assemblage, conservation concerns and river reach. Target species can be manually assigned to the highest sensitivity class, regardless of their original score, to account for regional conservation concerns and management targets.

The single elements of the EFHI will be briefly described in the following.

15.2.1 Flow Alterations

The EFHI assesses the impact of both upstream and downstream flow alterations. Upstream flow alteration is assessed based on the HPPs’ reservoir storage capacity, which is typically given relative to the average net inflow per time period (Langbein 1959, McMahon and Mein 1978). Reservoirs with a storage capacity exceeding the average annual net inflow are scored high risk. Smaller reservoirs and impoundments were further discriminated by their average flow velocity: Those reducing the mean flow velocity below 0.5 m/s were classified moderate risk and those still maintaining mean flow velocities of 0.5 m/s or higher low risk.

Downstream flow alterations are typically attributed to either hydropeaking or water abstraction (e.g., in residual flow stretches of diversion-type plants). Because hydropeaking inherently results in a completely altered discharge regime with severe impacts on stream biota, this operation mode was always scored high risk. The hazard of water abstraction, particularly problematic in residual river stretches of diversion schemes, was scored by environmental flow recommendations following Tharme (2003). Remaining discharge below 10% of the mean annual low flow (MNQ) was scored high risk. Higher discharge, but less than or equal 10% of mean annual flow (MQ) was scored moderate risk and >10% MQ low risk. The overall downstream risk was assigned according to the higher risk class. Upstream and downstream hazard classes were aggregated into the total flow alteration hazard using the same principle.

Table 15.1 Weighed numerical impact scores for specific HPP hazards and sensitive species for calculating of the EFHI. Target species sensitivity scores from low (2) to high (4) from van Treeck et al. (2020)

Hazard classification	Target species sensitivity		
	4 (high)	3 (moderate)	2 (low)
High	1	0.75	0.5
Moderate	0.75	0.5	0.25
Low	0.5	0.25	0

15.2.2 Entrainment and Turbine Mortality (ETM)

The total flow rate of all installed turbines was used as proxy of the entrainment risk for fishes describing the probability of fishes passing through the turbines rather than taking any other route downstream. This risk was estimated as ratio of flow rate to mean discharge (MQ) and scored high at ratios ≥ 1 , moderate between >0.5 and <1 and low ≤ 0.5 .

To account for the fish-deflecting effect of installed fish guidance structures (FGS), the entrainment risk score was adjusted for each installed turbine and turbine mortality hazard score as follows: Width-to length-ratios of target species empirically derived by Ebel (2013) to estimate the maximum length of both eel-like and non-eel-like (e.g., fusiform) species that could physically pass the FGS at a given bar spacing.

The turbine-specific mortality risk was assessed based on three components: (i) model-based turbine blade strike rates for Francis and Kaplan type turbines, (ii) empirical mortality rates for Archimedic screws, Kaplan very-low-head, Ossberger, Pelton, Pentair Fairbanks and water wheels, and (iii) empirical turbine barotrauma mortality rates. Following Wolter et al. (2020), mortality rates between 0 and 4% we scored low, between $>4\%$ and 8% moderate and $>8\%$ high risk. To evaluate HPPs with more than one turbine, the EFHI assigns risk scores for each individual turbine and species as shown in Table 15.1 and, subsequently, aggregates the relative contribution of each turbine to the overall mortality rate weighted by the turbines' specific flow rates. The turbine with the highest flow rate gets the proportional relevance "1" and all further turbines an equal or proportionally lower proportional relevance depending on their flow rate. The proportional relevance of each turbine serves to weigh the mortality risk score.

We applied the frequently used blade-strike model by Montén (1985) that calculates the probability of fishes striking a blade depending (among others) on their length and the space between turbine blades. For these models we used either the body length of fishes being physically able to pass the FGS or common length of an adult specimen if no FGS was installed. Modelled mortality rates for Francis and Kaplan turbines were classified low (0–4%), moderate (>4 –8%) and high ($>8\%$), respectively.

Other turbines without available mortality models were scored based on empirical data. Low risk turbines were water wheels ($<1\%$ mortality, Schomaker and Wolter 2016), Pentair Fairbanks ($<1\%$ mortality, Van Esch and van Berkel 2015; Winter et al. 2012), and very-low-head turbines (VLH, $<4\%$, Hogan et al. 2014; Lagarrigue et al. 2008; Fraser et al. 2007). Of moderate risk were Archimedes screws ($<8\%$, Wolter et al. 2020) and of high risk Ossberger ($>99\%$, Gloss and Wahl 1983, Knapp et al. 1982) and Pelton turbines ($>99\%$, Cada 2001).

Modelled strike mortality rates were complemented with a risk assessment of pressure-related injuries derived from the HPP's hydraulic head. Barrier height < 2 m was scored low risk, 2–10 m moderate and >10 m high risk based on empirical data (Wolter et al. 2020). The hydraulic head risk score was further adjusted according to species-specific susceptibility to barotrauma and quantitative model observations by Wilkes et al. (2018).

The risk score was set to zero for species without swim bladder and thus, not experiencing barotrauma risk. The risk score remained unchanged for physostomous fishes with an open swim bladder allowing for quicker pressure compensation, and it was raised by 50% for physoclistous fishes with closed swim bladder unable to quickly balance pressure changes (Brown et al. 2013, 2014; Colotelo et al. 2012; Harvey 1963; Wilkes et al. 2018).

15.2.3 Upstream Fish Passage

A major driver of the effectiveness of upstream fish passage facilities is their discharge relative to the mean discharge of the river, with higher values increasing passage success (Wolter and Schomaker 2019). We used two linear regressions to determine the minimum recommended discharge in an upstream migration facility as a function of the discharge of the stream. One was applied to rivers with a mean discharge up to 25 m³/s and a discharge in the upstream migration facility between 3 and 5% of that value, while the other was used for larger rivers that need less proportional discharge in the upstream migration facility (between 1 and 3%). Input discharge values as well as calculated “best-practice” values were rounded to one decimal point and compared. If the actual discharge in the upstream migration facility equalled or exceeded recommendations, the risk class was scored low. A discharge $\geq 50\%$ of the calculated recommendation was scored moderate risk. Discharges $< 50\%$ of the recommendations or no upstream migration facility were both considered inadequate and scored high risk. The superior passage performance of nature-like fishways was acknowledged by lowering both the upstream and downstream passage score by 20% each (in combination with FGS preventing turbine passage).

15.2.4 Downstream Fish Passage

Hazards for downstream migrating fish were scored as follows: Angled bar racks, louvers, modified and curved bar racks at a horizontal installation angle of $\leq 45^\circ$, or vertically inclined bar racks with an inclination of $\leq 45^\circ$, all in combination with downstream bypasses accessible across the whole water column have proven efficient (Calles et al. 2013a, b; Ebel 2013) and scored low risk. FGS installed at larger horizontal or rather steep vertical angles are demonstrably less efficient. Although they still prevent fishes from entrainment, they also cause damages or mortality due to impingement or shear forces (e.g., Calles et al. 2013a, b, 2010; Larinier and Travade 1999). In addition, the efficiency of bypasses accessible only at discrete location in the water column is highly variable (Calles et al. 2013a, b; Gosset et al. 2005; Økland et al. 2019; Travade et al. 2010). Therefore, vertically inclined bar racks and horizontal FGS with angles $> 45^\circ$ as well as any constellation without fully accessible bypass were scored as moderate risk.

A missing FGS or bypass was scored as high risk. Similar to upstream fish passage hazards, the bi-directional performance of a nature-like fishway is rewarded by reducing the downstream passage hazard score by 20%.

15.2.5 Final EFHI Score

The calculations outlined above produce in total 20 single score values: four hazard components individually assessed for five fish species each. Their aggregation is conducted in two steps; first the arithmetic mean is calculated for each component and second, the component means are averaged to the final EFHI score. Final scores ≤ 0.33 were classified “low risk”, scores between >0.33 and 0.66 were classified “moderate risk” and scores >0.66 were classified “high risk”.

15.2.6 Application of EFHI

To demonstrate the EFHI, it was applied to a small hydropower installation (<1 MW) in southern Germany. The plant creates only a small impoundment, but the flow speed therein is relatively slow (<0.5 m/s). The scheme is a diversion plant, with a well water-supplied residual river stretch. It has one Kaplan turbine and two Archimedes screws installed. The Kaplan turbine is four-bladed with 2.5 m outer and 0.95 m hub diameter and 100 rounds per minute (RPM) rotational speed. The total flow rate of all three turbines is $36 \text{ m}^3/\text{s}$, of which $18 \text{ m}^3/\text{s}$ go through the Kaplan turbine and $9 \text{ m}^3/\text{s}$ each through the two screws. The Kaplan runner is equipped with a vertical bar rack with 20 mm bar spacing and the two screws are protected by a 150 mm trash rack. The upstream migration facility has $1 \text{ m}^3/\text{s}$ admission flow. A downstream bypass is not installed. The surrounding fish fauna is dominated by rheophilic cyprinids; the target species used in EFHI are chub *Squalius cephalus* (sensitivity 3.0), barbel *Barbus barbus* (4.1), minnow *Phoxinus phoxinus* (2.5), spirlin *Alburnoides bipunctatus* (2.1) and dace *Leuciscus leuciscus* (3.1). Applying the EFHI calculation software tool (available at <https://zenodo.org/record/4686531>) as briefly described above, this particular constellation of HPP and river characteristics, target fish species and conservation concerns yields an overall EFHI score of 0.51 indicating a moderate risk.

To enhance fish protection, the plant could be refurbished with a finer screen of 15 mm bar spacing only and a fully accessible downstream bypass. In addition, increasing the flow speed in the impoundment to >0.5 m/s would further improve the EFHI score. These measures together would reduce the overall EFHI score to 0.31. In contrast, if there would be no functioning upstream fish migration facility and the bar spacing of the fine screen widened to e.g., 30 mm, the same plant would be scored 0.58. These example calculations demonstrate the EFHI application to assess existing hydropower plants and

the effects of potential mitigation measures and modifications of plant components on the hazard risk for fishes.

The EFHI can also be used to compare risk variations of similar HPPs in response to differences in sensitivity and conservation value of the ambient fish assemblages. For example, if the same HPP characterized above is located in an European eel (*Anguilla anguilla*) catchment hosting more sensitive fish species (e.g., trout *Salmo trutta* and nase *Chondrostoma nasus* instead of minnow and spirlin), the higher conservation value of the fish assemblage would substantially raise the final EFHI score to 0.68 “high risk”, even though stream size and HPP components remain unchanged.

The hazard components and final EFHI scores of the four scenarios outlined above are displayed in Fig. 15.2.

15.3 Cumulative Impact Assessment

The larger the rivers are the more they become exposed to multiple pressures, which might antagonistically or synergistically interact and result in cumulative impacts of pressures. While a variety of pressures might create cumulative impacts, the following section addresses only cumulative impacts of a series of barriers with or without hydropower plants in a river. It aims to provide guidance that allows hydropower developers as well as water managers to consider effects at a catchment scale, and thus maintain a high level

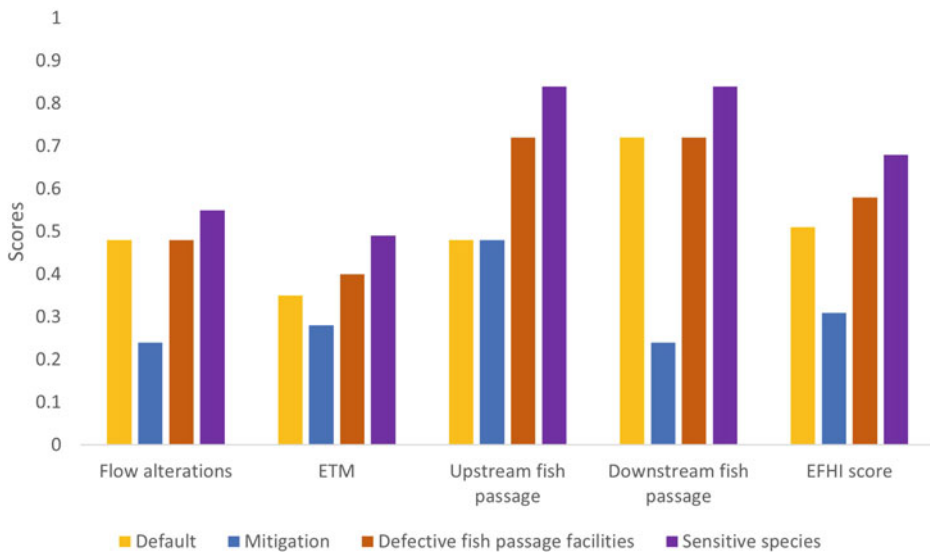


Fig. 15.2 EFHI scores of an exemplary small hydropower plant in default operation mode (yellow), after improving fish safety (blue), after failure of the fish protection measures (brown) and assuming a more sensitive ambient species community (purple)

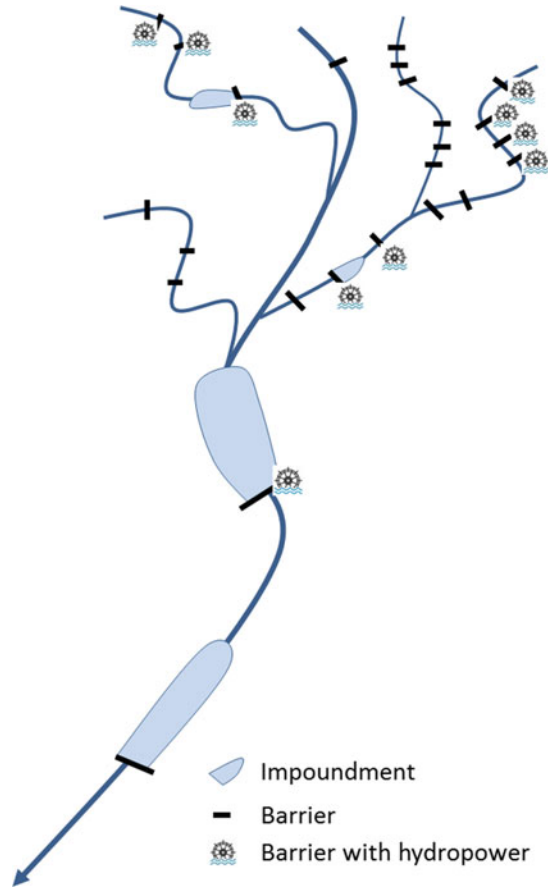
of environmental protection in line with the Water Framework and Habitats Directives. The cumulative impact assessment (CIA) presented here provides valuable inputs that need to be considered when a hydropower scheme is proposed and developed for construction or retro-fitted to mitigate potential impacts. However, it is important to recognize that generic models cannot be built for this purpose, because every scheme is different and its contributions to power production will vary depending on river conditions and operational practices. Whilst offering generic guidelines for assessment of cumulative impact of multiple hydropower schemes, CIA is limited in its assessment of risks and uncertainty about the impact of individual schemes on fisheries and the environment, which must be assessed using scheme-specific impact assessments and then used to understand the role of individual schemes in a wider cumulative assessment. The CIA protocol is constructed to support the Decision Support System for hydropower schemes that is presented in Sect. 15.4 of this chapter.

The CIA approach considers all transversal barriers in a river system. Depending on available migration facilities for fish, barriers form more or less significant migration obstacles for fish, fragment habitats and populations. Barriers are also obstacles for flow and sediment transport creating impoundments upstream with lower flow velocities and higher sedimentation, which typically result in habitat loss for rheophilic, gravel spawning river fishes. Therefore, the CIA approach also considers the cumulative length of impoundments in a river system, which corresponds to habitat loss for river fishes. It must be noted that numerous eurytopic and stagnant water-preferring species will benefit from impoundments. However, this benefit is not positively counted, because it does not correspond to the type-specific fish community and thus to a good ecological status of the river system according to the Water Framework Directive. Finally, some of the barriers are used for hydroelectricity production. Here in addition to the other impacts also fish mortality at hydropower plants might accumulate and is considered by the CIA. The different spatial elements considered in the CIA approach are illustrated in Fig. 15.3.

From a fish-ecological perspective it has to be considered, that diadromous species—these are species that obligatorily use marine and freshwater habitats during their life cycle—essentially have to pass all obstacles between their marine and freshwater habitats. Therefore, these species, e.g. European eel and Atlantic salmon, will experience the full cumulative effects of all pressures. Potamodromous species are also obligatory migrants, but they complete their life cycle only in freshwaters. Depending on habitat availability, they might pass only a limited number of barriers that will exert cumulative impacts.

The majority of non-obligatory or facultative migrating species does not depend on regular migrations and might complete their life cycle even within river fragments depending on habitat availability. However, all fishes show more or less extended exploratory movements and homing behaviour, with larger fishes moving longer distances and using larger home ranges (Radinger and Wolter 2014). Therefore, in particular the large-bodied species face a higher encounter probability with barriers and hydropower plants, but they hardly have to pass more than one barrier and thus will be least impacted by cumulative effects.

Fig. 15.3 Elements of cumulative impact assessment: cumulative share of impoundments/habitat loss for river fishes, cumulative number of barriers/fragmentation/migration obstacles, and cumulative mortality at hydropower plants



Independent of their migratory life history trait, all fish species will be affected by the cumulative length of impoundments that changes the hydromorphologic river character and by the fragmentation of habitats that increases with the number of barriers in a river system.

15.3.1 Cascades of Consecutive Impoundments

Generally, the natural power potential of Europe's watercourses is heavily utilized for energy production, particularly in alpine areas such as Norway, Austria, Switzerland or Southern Germany where topographical and hydrological conditions are most suitable. For example, the degree of hydropower development is already at a level of about 95, 90 and 70% in Switzerland, the Federal province Upper Austria and Bavaria, respectively (Jungwirth et al. 2003). At a global level, conservative estimates assume a total surface area of reservoirs of more than 400,000 km² (source: internationalrivers.org), which does

not include habitat losses generated by smaller dams and weirs. As a result, 15% of the global annual river runoff is stored in reservoirs (Likens 2010) and 48% of the global river volume is moderately to severely impacted by flow regulation, fragmentation or both (Schmutz and Moog 2018). Moreover, hydropower plants often occur in cascades creating a series of impounding reservoirs in one stream and thereby cumulatively contributing to the overall change of the hydraulic and habitat conditions of otherwise naturally free-flowing ecosystems.

Therefore, it is a main objective to quantify and assess the habitat loss caused by the cumulative impoundment effects of multiple anthropogenic barriers (i.e. cascades of dams and weirs) that alter the natural flow and habitat conditions of rivers with impacts on the riverine fish community. This involves several methodological questions related to the estimation/quantification of the spatial effects of impoundments (e.g. how large are impoundments and how much habitat area is altered) as outlined below.

Delineating Impoundments

A rather obvious way to delineate river impoundments is their direct on-site measurement in the field. While this might be considered time and resource consuming, it is very accurate to map flow modifications on-site, i.e. to assess deviation from natural/reference flow conditions and to determine the spatial extent of the impoundment in an upstream direction, especially in smaller streams. For example, to facilitate the assessment of flow modifications in Germany, the State Agency for Nature, Environment and Consumer Protection (LANUV) provided an overview of reference conditions of naturally flowing rivers based on hydromorphological river typologies (Timm et al. 1999).

Alternatively, the impoundment effect can also be roughly calculated from the dam height and the natural stream slope. Therefore, the dam height, HD is simply divided by the bottom slope S:

$$\Delta x = HD/S$$

For example, a 1 m high dam in a stream with a natural slope of 2 ‰ (0.002) creates an impoundment of about 500 m length.

Impact Assessment of Habitat Loss Due to Impoundments

We consider the total sum of lengths of all impounded sections of a river relative to the remaining free-flowing sections as the most decisive measure of the cumulative effect size of impoundments. Therefore, the cumulative length of impoundments per river section is divided by the total length of that river section. As a section, we consider the segment of the river between two confluences, i.e. points where two rivers merge. This follows the general principle of portioning streams into segments, which is also used in classical stream order concepts. This assures that relatively small impoundments of smaller streams are not compared to the entire (potentially free-flowing) main stem which generally provides habitat for different fish communities. Vice versa, habitat losses due to

impoundments in the main stem (or higher order rivers) should be addressed independent of (potentially free-flowing) head water sections, to account for the fact that different parts of the river along the longitudinal gradient are inhabited by different fish communities.

To provide an example, we followed the outlined approach to calculate the relative length of all impoundments per river section (= cumulative length of impoundments of river section i /total length of river section i) for all Austrian rivers. Similarly, we calculated the relative barrier density (= number of barriers of a river section i /total length of river section i). Spatial data on impoundments, barriers and the watercourse were obtained from the Water Information System Austria (WISA, <https://maps.wisa.bmlrt.gv.at/gewaesserbewirtschaftungsplan-2015>). Vector maps indicate in-situ mapped impoundments. River sections were extracted as lines between two confluences. For the visual analysis relative share of impoundments per river section as well as barrier density data were grouped into classes of stream order ranging from headwater streams (classes 1–3) to large lowland rivers (class 9, River Danube). The results show that the relative cumulative share/length of impoundments is increasing with increasing stream order (Fig. 15.4) while the barrier density is highest in low order streams and decreasing downstream. This becomes especially evident for the River Danube (stream order = 9) where most river sections are already impounded and only a few free-flowing river sections remained, whereas the number of barriers (i.e. barrier density) is comparably low. By comparison, sections of lower order head water streams might have higher numbers of small weirs and thus many impounded sections; however, these impoundments are usually rather short and there are still many non-impounded river sections.

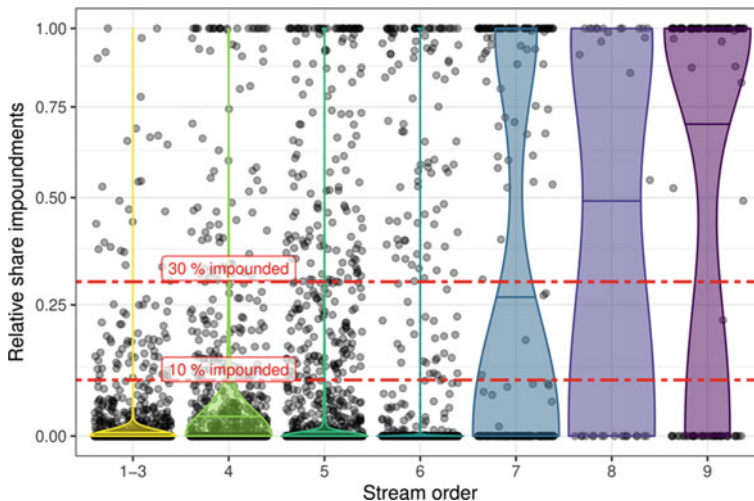


Fig. 15.4 Relative cumulative length of impoundments per river section and across stream orders in Austria (data from Water Information System Austria). Left to right is the upstream–downstream gradient. Points show relative cumulative impoundment length for single river sections. Violin plots indicate density distribution for a given stream order class

15.3.2 Fragmentation Due to Multiple Barriers

In addition to the alteration of river habitats due to impoundments, hydropower and their associated dams and weirs fragment river networks. Barriers represent one of the largest anthropogenic impacts on river ecosystems and limit habitat connectivity at multiple spatial and temporal scales (Fuller et al. 2015). This might express in particular in the impediment of ontogenetic migrations (e.g. spawning runs) and ordinary habitat movements of river fish (e.g. Marschall et al. 2011; Radinger and Wolter 2015; Radinger et al. 2018) and associated genetic fragmentation of populations (e.g. Gouskov et al. 2016). Recent studies emphasized that the location of a barrier within a river system and especially its location relative to suitable habitats and species occurrences determines its impact on fish (Kuemmerlen et al. 2016, Radinger and Wolter, 2015). Nevertheless, even moderate densities of barriers might not be acceptable given the commonly high mobility of river fish (Radinger and Wolter 2014). For example, Radinger et al. (2015) modelled the occurrence patterns of riverine fish in response to hydromorphological variables and in-stream structures and gained best modelling results when the conditions in a distance of 1–4 kms up- and downstream of a site were considered, thus indicating potentially extensive movements of fish within and between habitats.

Corresponding to previous studies (e.g. Van Looy et al. 2014), we consider the number of barriers per river km (i.e. relative barrier density) a suitable and easily obtainable indicator of the degree of fragmentation of a given river section. Analogous to the calculation of the relative share of impoundments, we consider a river section as the segment of the river between two confluences, i.e. points where two rivers merge. This assures that parts of the river network with rather low densities of barriers, as common in low-gradient streams and rivers, are not mixed with highly fragmented river network sections, as more common in high gradient streams as illustrated in Fig. 15.5 for Austrian rivers.

Assessment of Habitat Loss and Fragmentation Due to Multiple Cumulative Barriers

Based on the median values (half of the river sections per stream order), the cumulative length of impoundments was rather low in low order river sections and increased downstream (Fig. 15.4). This relation very well reflects the hydromorphic conditions of predominantly Alpine and Prealpine river systems, with steep slopes resulting in low impoundment length. However, the cumulative length of impoundments is a metric for habitat loss only and does not allow conclusions on the severity of habitat fragmentation and migration barriers. Inversely to the cumulative length of impoundments, the relative number of barriers per river kilometre was high in low order river sections and decreased in the downstream direction (Fig. 15.5).

These findings lead to a scoring of impacts and habitat loss resulting from impoundments and fragmentation due to barriers as shown in Table 15.2.

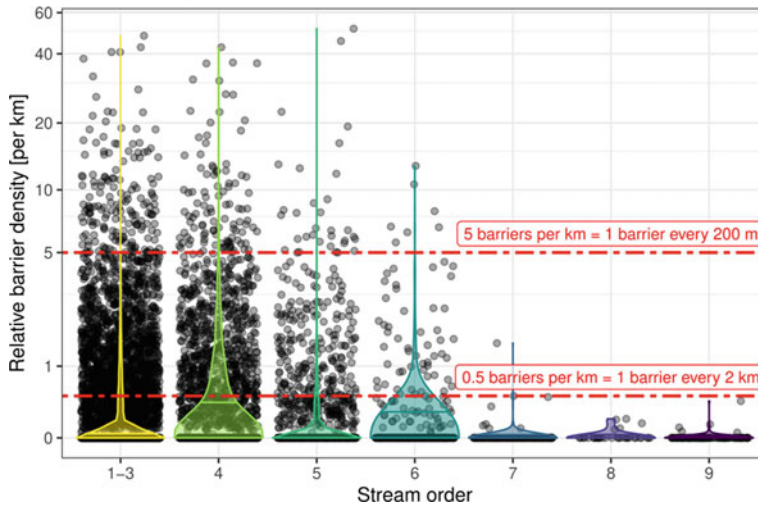


Fig. 15.5 Relative barrier density (number of barriers per km) per river section and across stream orders in Austria (data from Water Information System Austria). Left to right is the upstream–downstream gradient. Single points show relative barrier density for a single river section. Violin plots indicate the density distribution for a given stream order class

Table 15.2 Impact scoring of habitat loss due to relative cumulative impoundment lengths and barrier density

	Low impact	Moderate impact	High impact
% impounded length (relative to the total length of the river section)	≤10%	>10–30%	>30%
Relative barrier density (barriers per km, per river section)	<0.5 barriers/km	0.5–5 barriers/km	>5 barriers/km

Given that the highest hydromorphic and ecologic state of a river is free flowing with functioning sediment generation, transport and sorting processes and a nearly undisturbed riverine species community, a slight deviation might be 10% habitat loss. Therefore, habitat losses up to 10% in total might be considered of low cumulative impact. Accordingly, habitat losses of up to 30% in total in a river segment might be accepted as moderate cumulative impact, while higher losses of habitats have a high impact on riverine species communities. The high cumulative impacts from habitat losses in most segments in higher order rivers corresponds well with the reported habitat degradation and deviation from good ecological status according to the WFD.

Given the high mobility of riverine fish, a barrier every two kilometres or less (i.e. 0.5 relative barrier density) is considered moderate fragmentation impact. Barrier densities of five (=1 barrier every 200 m) clearly conflicts with the movement behaviour and

home range of most river fish (Radinger and Wolter 2014) and thus is considered a high fragmentation impact.

15.3.3 Cumulative Impacts—Diadromous Species

Understanding and evaluating cumulative impact of multiple HPPs in a catchment context requires fitting the life history characteristics to a life cycle model of the target species. These vary for different migratory species groups and account for the impact of individual schemes on components of the population life cycle.

Conceptual frameworks to assess the cumulative impact of multiple hydropower schemes are provided in Figs. 15.6 and 15.7 for anadromous and catadromous species, respectively. Each framework provides a starting point for assessment based on a count of the original adult population and follow this through the life cycle determining where the population will be impacted and by what proportion in addition to natural mortality. The anadromous framework starts with upstream migration passed the scheme (upstream barrier effect), opportunity for spawning and recruitment to the extant population based on habitat availability and finally losses caused by downstream migration through the successive hydropower schemes (Fig. 15.6). The catadromous framework (Fig. 15.7) starts with adults departing the river habitats and accounts for losses caused by downstream

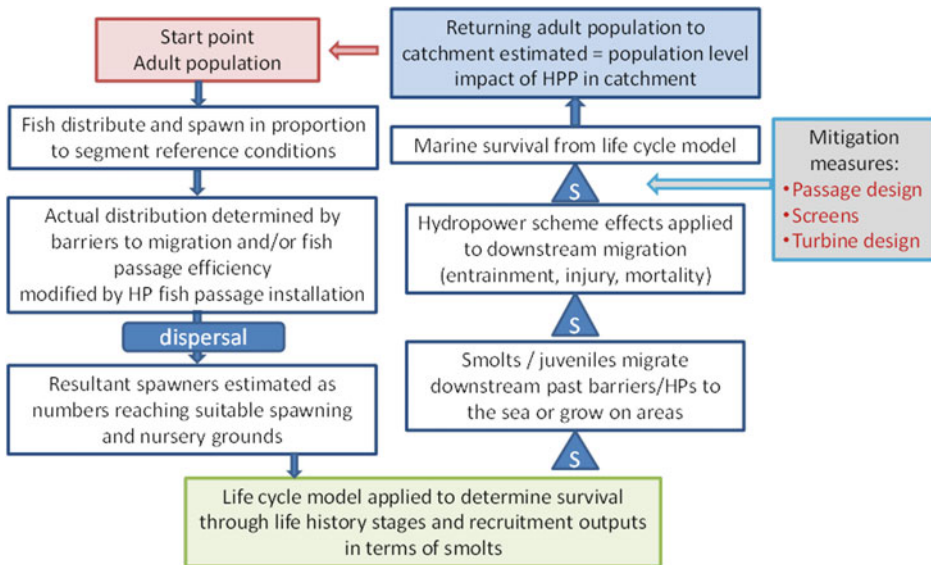


Fig. 15.6 Cumulative impact assessment for anadromous species—inputs and losses

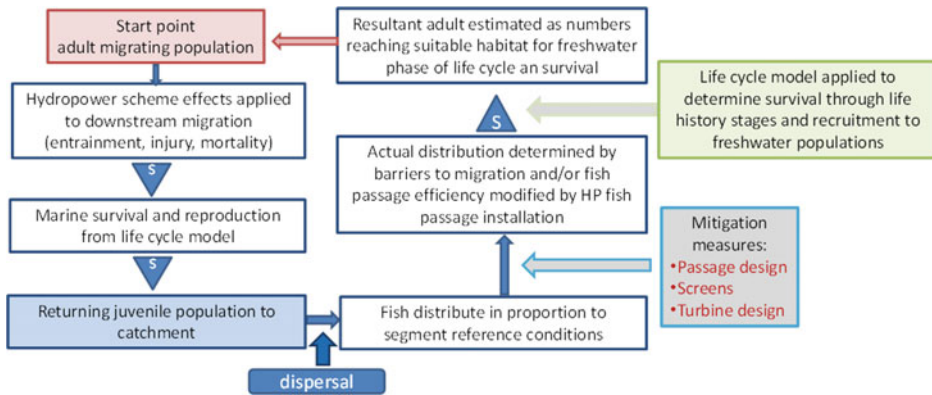


Fig. 15.7 Cumulative impact assessment for catadromous species—inputs and losses

migration through the successive hydropower schemes and then the impact on returning juvenile fishes dispersing in the target catchment following recruitment in the marine environment.

Each component of the life cycle can be modelled based on criteria specifically related to each individual scheme or based on field surveys. Where such data are not available, expert judgement or available information in the literature should be used to populate the models. Key elements in these frameworks are assessing impediment to upstream migration, loss of important habitat upstream of the hydropower installations and thus loss of recruitment to the extant population and loss of fish during downstream migration as a result of injury or mortality at the hydropower schemes or impacts of delayed migration. Each of these parameters can be quantified and the additive effect can be determined to understand the population impact of the cascade of hydropower schemes.

Barrier Passability

One of the key impacts of hydropower schemes is the disruption to connectivity caused by the dam structure. All hydropower schemes create a barrier to hold back or divert water to the turbine(s). The size of the barrier is highly variable depending on the design of the scheme but needs to be accounted for. An example of the cumulative impact of multiple barriers on a system is shown in Fig. 15.8. Here the impacts of seven barriers in succession on the population size of an upstream migrating species are compared with different levels of fish passability. It can be clearly seen that the cumulative effects of compromised passabilities < 0.5 (i.e. less than 50% of the approaching specimens successfully passed) at the barriers result in extirpation of the population in the upstream areas. It is thus essential to model the impact of variable passabilities at barriers to determine the cumulative impact.

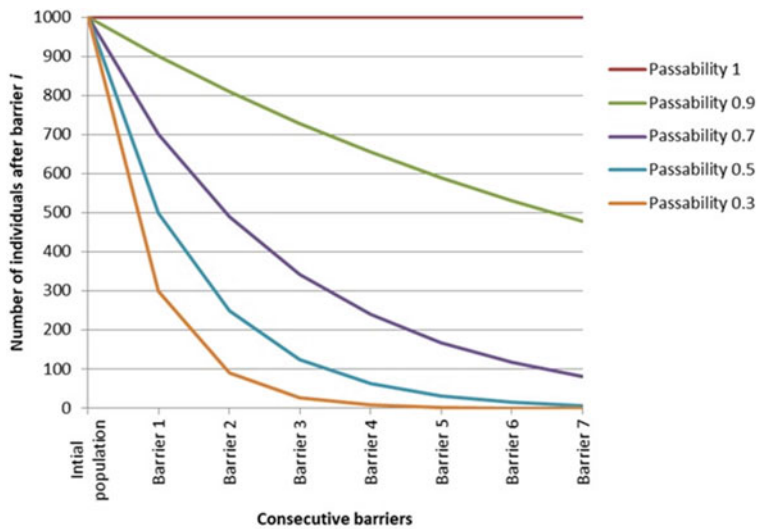


Fig. 15.8 Cumulative impact of multiple (seven) consecutive barriers of a given passability rate (from 0.3 = 30% of approaching fish to 1 = 100%)

A number of tools are available to assess barrier passability for fish (Kemp and O’Hanley 2010). A coarse resolution rapid barrier assessment methodology that is suitable for multiple fish species and considers both up and downstream dispersal was devised by Kemp et al. (2005) and later in 2012, by the Scotland and Northern Ireland Forum for Environment Research (SNIFFER). The assessment method uses rule-based criteria for fish morphology, behaviour, swimming and leaping ability to estimate barrier passability. The latter is the fraction of fish (in the range 0–1) that are able to successfully pass a given barrier. Each barrier is assigned one of four passability levels as follows: 0 is a complete barrier to movement; 0.3 is a high impact partial barrier, passable to a small proportion of fish or passable only for short periods of time; 0.6 is a low impact partial barrier, passable to a high proportion of fish or for long periods of time; and 1 is a fully passable structure.

Downstream Migration

Once fish have spawned, adults of many species must migrate downstream, either to the sea (anadromous species) or the lower reaches of the river (potamodromous species). The juveniles of both migratory guilds will also ultimately migrate downstream to complete their life cycles. In addition, adult catadromous species such as eel must eventually migrate downstream to complete their life cycles. In European rivers downstream migrating fish mostly actively swim downstream and may pass the dam by one of several corridors: downstream migration facility, turbine, water release over spillway or through sluice gates. In addition, it should be recognised that some juvenile life stages of fish move

downstream by drifting in the current. This mode of migration past hydropower structures is dependent of the allocation of flows and with the majority water going through the turbines, most juveniles will experience the trauma of high pressure and sheer and will unlikely survive the experience.

As indicated, one route for downstream fish migration is passage through the turbine at hydropower installations, which can be related to injury and mortality, caused by several damage mechanisms as already outlined for the EFHI and in Chap. 4. Harrison et al. (2019) posit that overall passage efficiency for diadromous fish populations using turbine routes to pass hydropower dams in a downstream direction (DS Passage efficiency) can be considered a product of the conditional probability of reservoir, forebay and turbine entry, along with turbine passage survival, turbine passage exit survival, delayed survival and sub-lethal effects. These effects are again cumulative and result in an almost complete loss if downstream passage is low and mortality at the power station is high (Fig. 15.9), which is often the case.

15.3.4 Cumulative Impacts—Potamodromous Species

Addressing cumulative impact for potamodromous species follows a similar approach to diadromous species except that the species of concern only migrate within the river system, either up and downstream including into tributaries or onto floodplain systems to complete their life cycles. The schematic framework of cumulative impact assessment for potamodromous species is shown in Fig. 15.10 and the models can be populated based on information provided in the following sections on successive dam passage, home range and mobility and migration distances.

Successive Dam Passage

Few studies have investigated the passage success of non-diadromous species over more than one consecutive barrier in either an upstream or downstream direction.

Fig. 15.9 Scheme for cumulative downstream passage efficiency for different mortality rates

	Mortality of 2%	Mortality of 70%
	START: 100% of all downstream migrating fish	
	⇩ Dam 1 ⇩	
	98%	30%
	⇩ Dam 2 ⇩	
	96%	9%
	⇩ Dam 3 ⇩	
	94%	2.7%
	⇩ Dam 4 ⇩	
	92%	0.81%
	⇩ Dam 5 ⇩	
	90%	0.24%

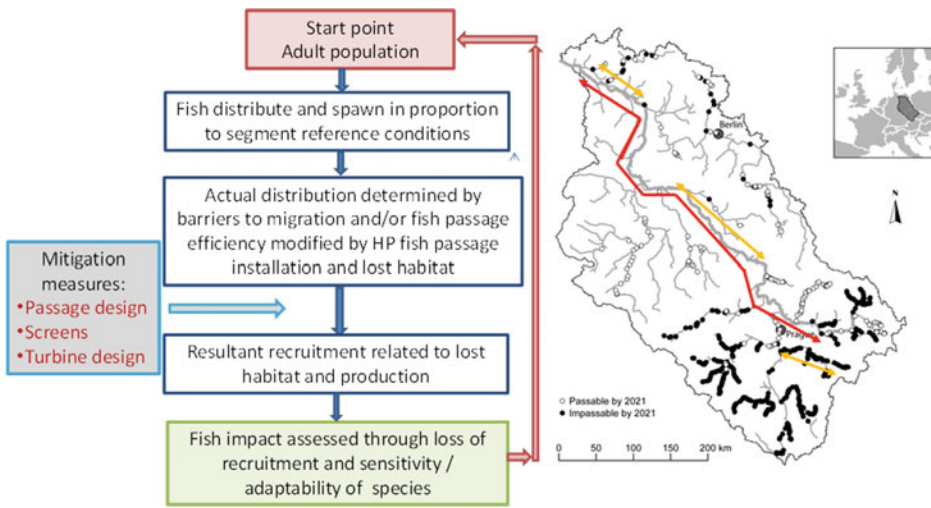


Fig. 15.10 Cumulative impact assessment for potamodromous species—inputs and losses

In the Dutch lower Rivers Rhine and Meuse, De Leeuw and Winter (2008) studied the movement of potamodromous fish species using transponders and a network of 29 detection stations. In total, 110 ide, 76 barbel, 51 chub and 8 nase were tagged, of them 57, 30, 15 and 4 specimens, respectively, were never detected during the study period between February 2003 and June 2006. Passage over weirs was observed for very few individuals only, as follows: of 2 approaching barbel 2 successfully passed the weir in an upstream direction, as did 3 of 4 chub, 1 of 2 nase, and 6 of 30 ide. Passage success in downstream direction was much less: 1 of 7 approaching barbel, 1 of 1 nase, 3 of 11 ide and not a single chub (De Leeuw and Winter 2008). The movement and passage over consecutive weirs of these four potamodromous species has also been studied by Benitez et al. (2018) in the Rivers Meuse and Ourthe in Belgium and by Lucas and Frear (1997) in the River Nidd (UK). All studies were performed using active or passive telemetry techniques and observed an average cumulative passage success of these species of 41% at the first barrier and 10% at the second.

Based on the little data available, there is evidence that migration of potamodromous cyprinids may fall to as low as 10% after bypassing two weirs. It has to be noted; however, that there might be less individual motivation in potamodromous fish for migrating larger distances compared with diadromous species, which also strongly depends on habitat quality and availability in the river reach between barriers (Benitez et al. 2018). It still remains challenging to identify the number of mobile specimens willing to pass longer distances over several barriers. More research is needed to gather information.

Migration Distance

To understand how far potamodromous species potentially migrate within river systems, empirical data on maximum migration distances have been compiled. They might serve as proxy to assess the maximum number of barriers a potamodromous fish has to pass during its life to complete its life cycle. The usual home range is typically much shorter. These migration distances vary according to river length and do not indicate long- or short-distance migrating as a kind of ecological trait of a species. Migrations have to cover the distance to the—not necessarily nearest (compare Fredrich et al. 2003, for chub *Squalius cephalus*)—spawning site or seasonal habitat. The spatial arrangement of river segments, river reaches and habitats strongly depends on river size and length. Accordingly, migration distances correlate with river size measured for example as Strahler or Shreve order. In larger rivers, the same species moves over significantly longer distances; therefore, river size and study time emerged as significant predictors of movement distance (Radinger and Wolter 2014).

Migration distances vary considerably within species depending upon individual motivation (Table 15.3). Fish personality, as well as genetic factors, seasonality, and purpose of the migration influence the distances covered (Brodersen et al. 2008; Brönmark et al. 2013). Resource migrations and diurnal and seasonal habitat shifts largely depend on the home range size (Fredrich 2003) and are thus related to river size (Radinger and Wolter 2014).

Also, discharge was reported to affect the annual migration distances as well as the intensity of spawning runs based on catch data for European sturgeons in the River Rhine (Kinzelbach 1987) and Atlantic salmon in the River Elbe (Wolter 2015). Especially in years with high floods, salmon were reportedly caught much further upstream than commonly observed (Wolter 2015). Similarly, further migration distances and improved passage of barriers must be assumed for potamodromous fish at higher discharges.

All potamodromous species are iteroparous, i.e. spawn several times during their reproductive life, which means that repeated upstream and downstream migrations for spawning are the rule. Typically, the downstream migration of the adults after spawning has no discrete runs or migration peaks but spreads over a longer period. Examples of downstream migration distances for potamodromous species are provided in Table 15.4.

Downstream migrations also take place when fish change sites to reach more productive or more suitable feeding grounds. Displacement through high water conditions or while searching for food may result in large scale downstream displacement. The same mechanism is effective during drift of larval fish or the undirected outmigration of juveniles. This movement is characterized by a non-uniform orientation towards the current which largely contrasts with active movement with the flow that is observed during adverse conditions (Lucas and Baras 2001).

Table 15.3 Empirical upstream migration distances of potamodromous and facultative migrating species

Species	Distance (km)	River	Method	Reference
Potamodromous migrants				
<i>Aspius aspius</i>	166	Elbe	Telemetry	Fredrich (2003)
<i>Barbus barbus</i>	22.7	Ourthe, Belgium	Telemetry	Ovidio et al. (2007)
<i>Barbus barbus</i>	12.7	Ourthe, Belgium	Telemetry	Ovidio et al. (2007)
<i>Barbus barbus</i>	>25	Meuse, Belgium	Telemetry	De Leeuw and Winter (2008)
<i>Barbus barbus</i>	2–20	Nidd/Ouse, England	Telemetry	Lucas and Batley (1996)
<i>Barbus barbus</i>	20	Nidd, England	Telemetry	Lucas and Frear (1997)
<i>barbus barbus</i>	34	Severn, England	Mark-recapture	Hunt and Jones (1974)
<i>Barbus barbus</i>	318	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Barbus barbus</i>	2	Jihlava, Czech Republic	Mark-recapture	Penaz et al. (2002)
<i>Chondrostoma nasus</i>	>25	Meuse, Belgium	Telemetry	De Leeuw and Winter (2008)
<i>Chondrostoma nasus</i>	140	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Leuciscus cephalus</i>	169	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Leuciscus cephalus</i>	>25	Meuse, Belgium	Telemetry	De Leeuw and Winter (2008)
<i>Leuciscus cephalus</i>	25	Spree	Telemetry	Fredrich et al. (2003)
<i>Leuciscus idus</i>	>45	Meuse, Belgium	Telemetry	De Leeuw and Winter (2008)
<i>Leuciscus idus</i>	19	Elbe	Telemetry	Kulíšková et al. (2009)
<i>Leuciscus idus</i>	>100–187	Elbe	Telemetry	Winter and Fredrich (2003)
<i>Leuciscus idus</i>	82	Danube	Mark-recapture	Steinmann et al. (1937)

(continued)

Table 15.3 (continued)

Species	Distance (km)	River	Method	Reference
<i>Leuciscus leuciscus</i>	0.03–0.7	Frome, England	Telemetry	Clough and Ladle (1997)
<i>Leuciscus leuciscus</i>	<0.2	Frome, England	Telemetry	Clough et al. (1998)
<i>Leuciscus leuciscus</i>	3.3	Frome, England	Telemetry	Clough and Beaumont (1998)
<i>Leuciscus leuciscus</i>	10–21			Lucas and Baras (2001)
<i>Lota lota</i>	128	Kootenai, USA	Telemetry	Paragamian and Wakkinen (2007)
<i>Lota lota</i>	157	Elbe	Mark-recapture	Faller and Schwevers (2012)
<i>Lota lota</i>	85	Tanana, Alaska	Telemetry	Breaser et al. (1988)
<i>Lota lota</i>	100	Elbe	Telemetry	Fredrich and Arzbach (2002)
Facultative migrants				
<i>Abramis brama</i>	<2–5.2	Trend, England	Telemetry	Lyons and Lucas (2002)
<i>Abramis brama</i>	58	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Esox lucius</i>	>50	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Esox lucius</i>	14.4	River Rena, NO	Telemetry	Sandlund et al. (2016)
<i>Gobio gobio</i>	>9.5	Donau/Melk, Austria	Mark-recapture	Zitek and Schmutz (2004)
<i>Sander lucioperca</i>	326	Elbe	Mark-recapture	Faller and Schwevers (2012)
<i>Sander lucioperca</i>	122	Elbe	Telemetry	Nellen and Kausch (2002)

Table 15.4 Empirical downstream migration distances of potamodromous and facultative migrating

Species	Distance (km)	River	Method	Reference
<i>Aspius aspius</i>	166	Elbe	Telemetry	Fredrich (2003)
<i>Aspius aspius</i>	125–164	Elbe	Telemetry	Fredrich (2003)
<i>Aspius aspius</i>	>100	Elbe	Telemetry	Fredrich (2003)
<i>Aspius aspius</i>	190	Elbe	Telemetry	Nellen and Kausch (2002)
<i>Barbus barbus</i>	22.7	Ourthe, Belgium	Telemetry	Ovidio et al. (2007)
<i>Barbus barbus</i>	12.7	Ourthe, Belgium	Telemetry	Ovidio et al. (2007)
<i>Barbus barbus</i>	22	Severn, England	Mark-recapture	Hunt and Jones (1974)
<i>Barbus barbus</i>	295	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Barbus barbus</i>	1.7	Jihlava, Czech Republic	Mark-recapture	Penaz et al. (2002)
<i>Chondrostoma nasus</i>	446	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Leuciscus cephalus</i>	15	Spree	Telemetry	Fredrich et al. (2003)
<i>Leuciscus idus</i>	>200	Meuse, Belgium	Telemetry	De Leeuw and Winter (2008)
<i>Leuciscus idus</i>	68–100	Elbe	Telemetry	Kulíšková et al. (2009)
<i>Leuciscus idus</i>	>100–187	Elbe	Telemetry	Winter and Fredrich (2003)
<i>Leuciscus idus</i>	105	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Leuciscus idus</i>	150	Elbe	Telemetry	Fredrich (2000)
<i>Leuciscus idus</i>	100–165	Elbe	Telemetry	Nellen and Kausch (2002)
<i>Leuciscus leuciscus</i>	0.03–0.7	Frome, England	Telemetry	Clough and Ladle (1997)
<i>Leuciscus leuciscus</i>	4.5	Frome, England	Telemetry	Clough et al. (1998)
<i>Leuciscus leuciscus</i>	9.1	Frome, England	Telemetry	Clough and Beaumont (1998)
<i>Lota Lota</i>	68	Tanana, Alaska	Telemetry	Breaser et al. (1988)

(continued)

Table 15.4 (continued)

Species	Distance (km)	River	Method	Reference
<i>Abramis brama</i>	<2–5.2	Trend, England	Telemetry	Lyons and Lucas (2002)
<i>Abramis brama</i>	75	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Abramis brama</i>	172	Elbe	Telemetry	Nellen and Kausch (2002)
<i>Abramis brama</i>	20–130	Elbe	Telemetry	Nellen and Kausch (2002)
<i>Esox lucius</i>	>50	Danube	Mark-recapture	Steinmann et al. (1937)
<i>Esox lucius</i>	14.4	River Rena, NO	Telemetry	Sandlund et al. (2016)
<i>Sander lucioperca</i>	35	Elbe	Mark-recapture	Faller and Schwevers (2012)

15.3.5 Managing Cumulative Effects

Cumulative impacts of transversal barriers on biota can be assessed even without particular information on the ecological status of riverine biota, solely based on the cumulative fragmentation/barrier density and the cumulative length of impoundments relative to the river length.

Both fragmentation and impoundments pose significant hydromorphic degradation resulting in the deterioration of the ecological status or potential of a river. Based on the significance thresholds suggested here (Table 15.2):

- i. new projects have to demonstrate that they will not raise the total length of impoundment or fragmentation above the respective thresholds (30% impounded and 5 barriers/km, respectively). In conservation areas, the stronger low-impact-thresholds should be applied (Table 15.4);
- ii. barriers might be selected and prioritized for removal to drop the total impounded river length below 30% or the number of barriers below 5/km, which will result in significant habitat gain for riverine biota including also non-migratory and facultative migrating fish species.

Cumulative impacts could be also assessed and managed only for selected species of conservation concern, e.g. for eels or migratory salmonids. This is the most elaborated application supported by predictable passage needs of diadromous species. Common management actions include implementation of upstream fish migration and downstream fish

protection facilities at barriers. The overall performance in mitigating cumulative impacts is determined by the efficiency of the individual facilities with the least performing ones causing the greatest bottlenecks. Therefore, the management of cumulative impacts is often reduced to identifying and mitigating the largest bottlenecks.

New hydropower projects at existing barriers have to outline already in the application phase how they will improve up- and downstream fish passage and protection and to show that they will not worsen the overall situation and dilute the population of the target species.

It must be noted that up- and downstream migration facilities can be effective for lampreys and fishes, but do neither enhance the hydromorphic quality of rivers and river processes nor other riverine biodiversity. The gold standard should be reducing the cumulative impacts of barriers on habitat quality and fragmentation below the outlined thresholds.

Much of the cumulative impact assessment at hydropower schemes described previously is based on the principle that the dam acts as barrier impeding upstream and downstream migration, modifies habitat quality up- and downstream to some degree and potentially causes mortality. In reality, these features vary between schemes depending on location, size, scale, infrastructure and hydrological and hydraulic characteristics of the installation. The precise information for each scheme cannot be accessed in a generic manner using models or similar tools, and must be collected for each hydropower scheme in a systematic manner.

As such, cumulative impact assessment serves to determine the overall impact of existing barriers and hydropower schemes in a catchment. The procedure also allows to determine effects of mitigation measures. For example, if a fishpass is newly installed or retrofitted for more efficiency, the gain for fish protection can be estimated using the EFHI. Similarly, installing turbines that are less damaging to fish or more protective screens will mitigate downstream mortality of fish.

Where a series of hydropower schemes are installed, the cumulative impact assessment framework enables analysis of which structure would provide the greatest response if specific mitigation measures will be applied. This can be linked to a cost–benefit analysis and the schemes can be prioritized according to the best return on investment.

Overall the perspective of cumulative impacts of multiple hydropower schemes in the same catchment must be accounted for in the regional planning context and mitigation measures to minimise their impacts built into the catchment planning procedures as well as measures against the loss of other services as a result of the multiple schemes. The CIA framework provided here allows more informed decision-making and prioritisation.

15.4 Decision Support Tool

15.4.1 Decision Making in Hydropower Development and Mitigation

The proliferation of hydropower development to meet obligations under the Renewable Energy Directive has also seen the emergence of conflict between the hydropower developers and the fisheries and conservation sectors. The hydropower industry wants a joined up response to their planned developments with sound guidance and robust decision-making. On the other hand, the fisheries and conservation interests have concerns over the impact that these schemes can have (especially the impact on WFD and Habitats Directive status) and want to consistent and robust evaluation of proposed schemes. They need to be reassured that the standards of design, construction and operation will provide adequate protection of wildlife and biodiversity, and the ecosystem services that they provide. As a result, there is a need to develop robust and transparent, evidence-based, support for decision making that is easy for developers and regulators to use while enabling a high level of appropriate environmental protection and mitigation.

Like all development projects, planning a new hydropower proposal or refitting an existing scheme involves consideration of a number of distinct stages within the project life cycle (for example see the Hydropower Sustainability Assessment Protocol, IHA 2018). These stages move from the early stage screening of proposals and scoping of options, to detailed project preparation, implementation, operation and scheme maintenance. It is likely that this process also requires full Environmental Impact Assessment (EIA) as part of the planning cycle, depending on the scale of the scheme and the nature of the impacts identified in early stage screening. At each stage in the project cycle, there are important development decision points that direct project planning and progression to the next stage. Some of the critical decisions can be faced in the early stage of the project life cycle. In the case of hydropower development, these decisions relate to the screening of proposals to identify impacts and the scoping of appropriate and effective mitigation requirements.

It is this early stage screening that informs key decisions and acts to scope the requirements for both full EIA and detailed options appraisal for mitigation measure requirements and design. Whilst guidelines exist for assessing the sustainability of hydropower schemes or the need for retrofitting existing structures, they are often limited in their assessment of risks and uncertainty about the impact of schemes on fisheries and the environment. Few provide support for the decision-making process at each stage of evaluation. At each stage of the process, decisions have to be made about the acceptability of the risks and uncertainty of impacts of hydropower schemes, and the ability to manage those risks. Existing decision frameworks and planning protocols (e.g. IHA 2018) provide valuable inputs into key areas that need to be considered when a hydropower plant (HPP) or scheme is proposed and developed for construction or retro-fitted to mitigate potential impacts. These included:

- i. assessment of licensing requirements for permitting of hydropower plants (e.g. permitting of abstractions and impoundments);
- ii. assessment of ecological impacts, including fisheries and biodiversity;
- iii. checking the ecological status of water bodies and the pressures already imposed on fish populations/communities by obstructions and hydropower in River Basin Management Plans
- iv. identification of mitigation measures (especially fish passage) and steps to take in either permitting of a new hydropower scheme or the re-fitting of an existing scheme (including provisions on risk assessment);
- v. consultation and determination (approval or rejection) of application/project plan.

Given this, a planning and decision support framework was developed in FITHydro and was aligned to the principles of project management and EIA, providing a structured early stage screening and scoping tool to support hydropower project planning in relation to impacts on fish and their mitigation. This screening tool, based on a risk assessment framework, is described here. The FITHydro Decision Support System (DSS) is not meant to replace existing national frameworks or general assessment protocols like the Hydropower Sustainability Assessment Protocol, but is intended to support these frameworks and improve decision making in respect of mitigation for fish-specific issues.

15.4.2 FITHydro Decision Support System

The FITHydro DSS is a systematic screening and scoping assessment tool implemented as open access, web-based system. The DSS aims to guide environmentally sustainable hydropower production and promote safe fish passage and sustainable fish populations, through the evidence-based planning, development and operation of existing and new hydropower schemes, including the revision of permits. It was designed to draw on the outputs and innovations from various tasks within the FITHydro project and in particular, the tool was designed to integrate with this knowledge base implemented as FITHydro wiki (Fig. 15.11).

The planning and decision framework developed within the FITHydro DSS provides a systematic approach to decision-making regarding proposals for hydropower schemes and mitigation of their impacts on fish populations. Integral to the decision support framework are risk assessment procedures that aid the decision to proceed with a scheme, prioritise which hazards require mitigation and to scope how the likely impacts of schemes could be mitigated.

Here, we summarise the FITHydro decision support system workflow (Fig. 15.12) which enables operators and regulators to develop structured proposals for new HPPs, and to both, review and risk assess, those proposals whilst identifying appropriate mitigation measures to address the impact of both new and existing HPPs. The DSS framework

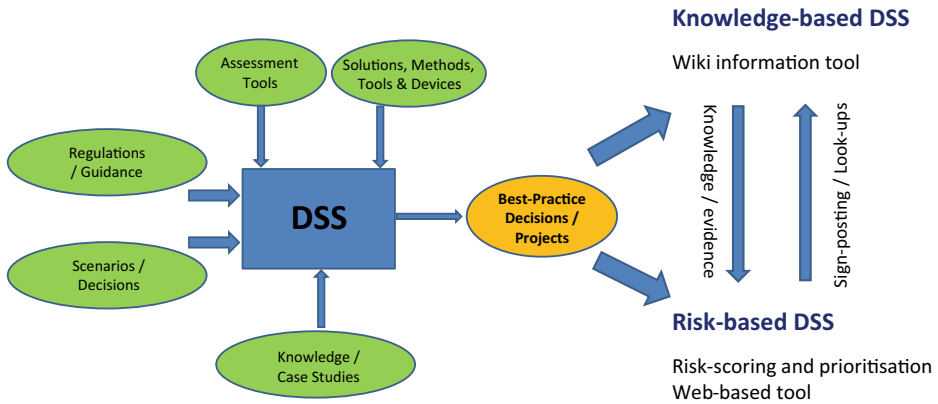


Fig. 15.11 Conceptual diagram for the elements of Decision Support Systems to ensure environmentally friendly hydropower decision making that underpin the concept for the FiTHydro DSS and selection of the most appropriate solutions, methods, tools and devices for mitigation of impacts

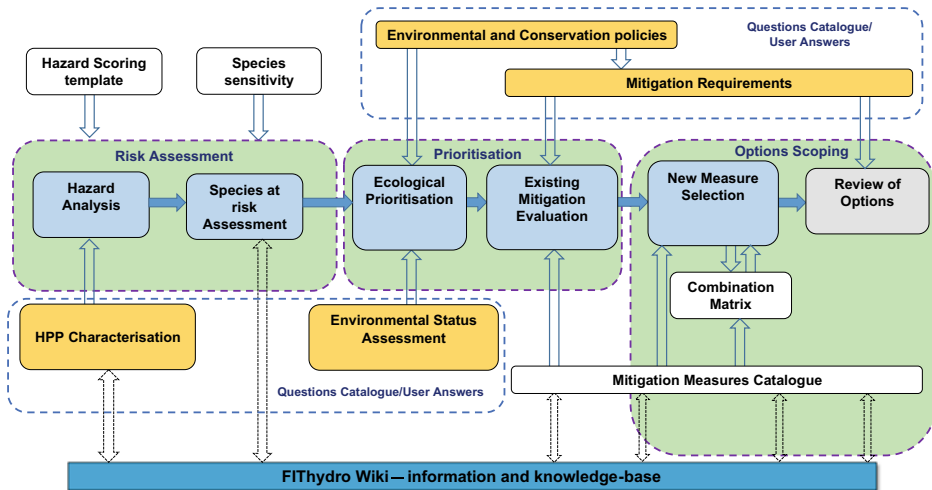


Fig. 15.12 Conceptual workflow diagram (left to right) of the Decision Support System—highlighting key steps (green), tasks/processes (blue), user inputs (yellow), system inputs/knowledge (white) and outputs (grey)

is sufficiently flexible to address decision making in two similar but contrasting scenarios (use cases):

- impact assessment and planning of new mitigation measures for existing HPPs (retrofitting).

- planning and risk assessment of a new HPP scheme proposal.

Broadly the overall decision process is the same for both scenarios, consisting of risk assessments, prioritisation and mitigation options scoping steps (Fig. 15.12). However, the first scenario necessitates that the existing mitigation measures at the scheme and their effectiveness are evaluated (Step 2 Prioritisation) prior to determining which pressures/impacts require mitigation. The framework (Fig. 15.12) leads the decision maker through a number of tasks, which act to characterise, risk-assess and prioritise the scheme(s), together with identifying most appropriate and potentially cost-effective mitigation options addressing the hazards and impacts arising due to the nature and context of the specific scheme(s).

The web tool implements question catalogues, risk assessment frameworks/matrices and database filtering tools which enables:

1. Pre-screening characterisation, hazard identification and risk assessment (Sect. 15.4.3, Fig. 15.13)
2. Ecological status assessment and review of existing mitigation (Sect. 15.4.4, Fig. 15.14)
3. Identification of appropriate mitigation measures and synergistic solutions (Sect. 15.4.5, Fig. 15.15)

Taken together, these three steps will produce a project screening/scoping report providing a systematic approach to decision-making for proposals relating to hydropower schemes and their effective mitigation.

The planning and decision support framework described here allows for proposals to be transparently and systematically evaluated at different levels and stages and provides a mechanism for the identification of the most appropriate and potentially cost-effective mitigation scenario addressing the pressures and impacts arising due to the nature and context of the specific scheme(s).

15.4.3 Scheme Characterisation, Hazard Identification and Risk Assessment

The impacts of hydropower on fish populations and communities have been reviewed extensively (e.g. Scruton et al. 2008, Wolter et al. 2019, Harper et al. 2020) and summarised elsewhere in this volume (see Chap. 4 and Sects. 15.2 and 15.3). Three main modes of impacts of the operation of hydropower on fish and the riverine environment have been identified (Harby et al. 2019):

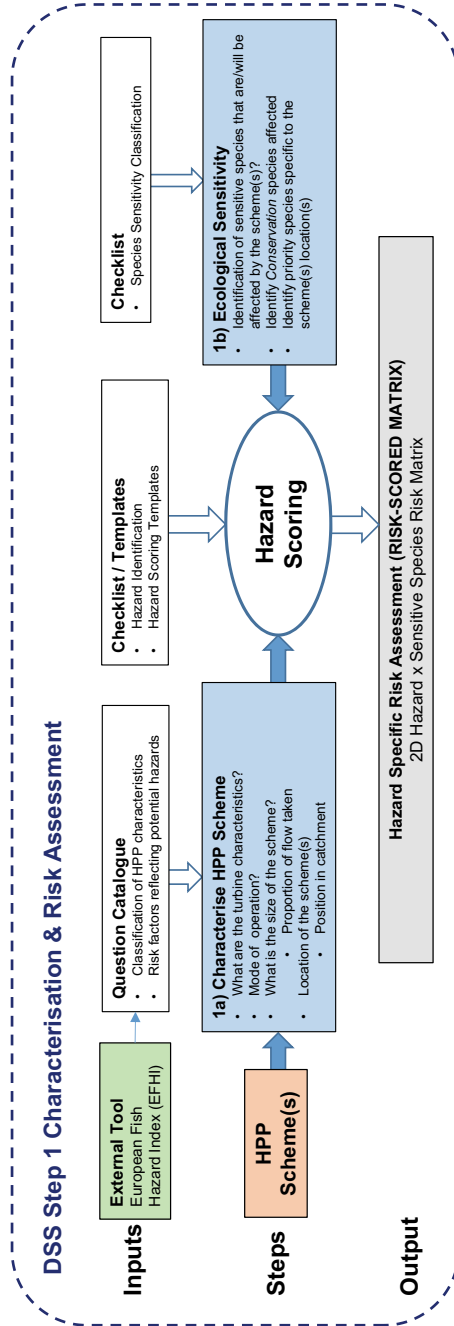


Fig. 15.13 Conceptual flow diagram of Step 1 of the Decision Support System Framework—pre-screening characterisation and risk identification. Green = external tools; blue = tasks; white = FTHydro inputs and matrices; grey = output

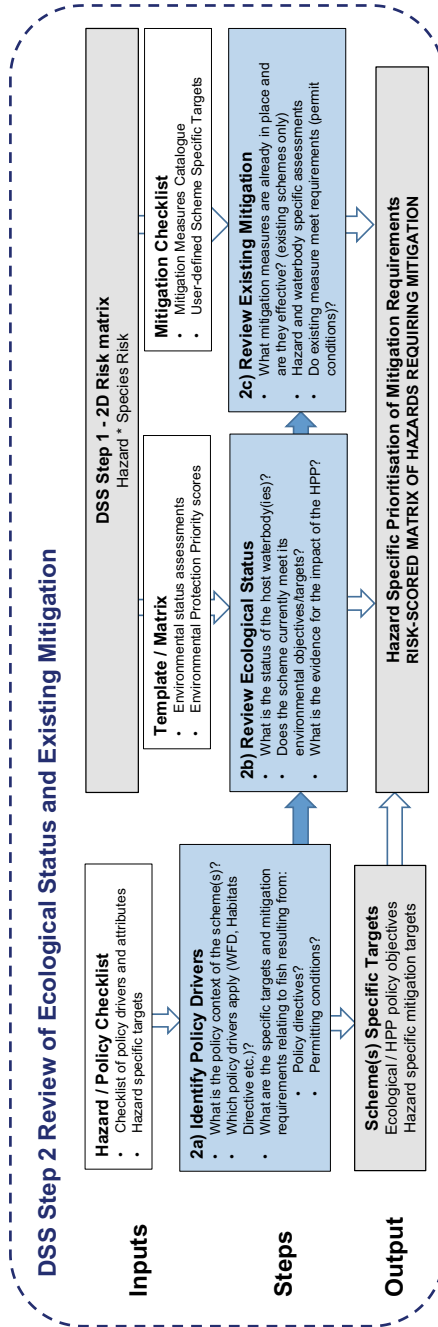


Fig. 15.14 Conceptual flow diagram of Step 2 of the proposed Decision Support System Framework—ecological pressures and impacts characterisation and risk identification. Grey = outputs/inputs to/from steps; blue = tasks; white = FIThydro inputs and matrices

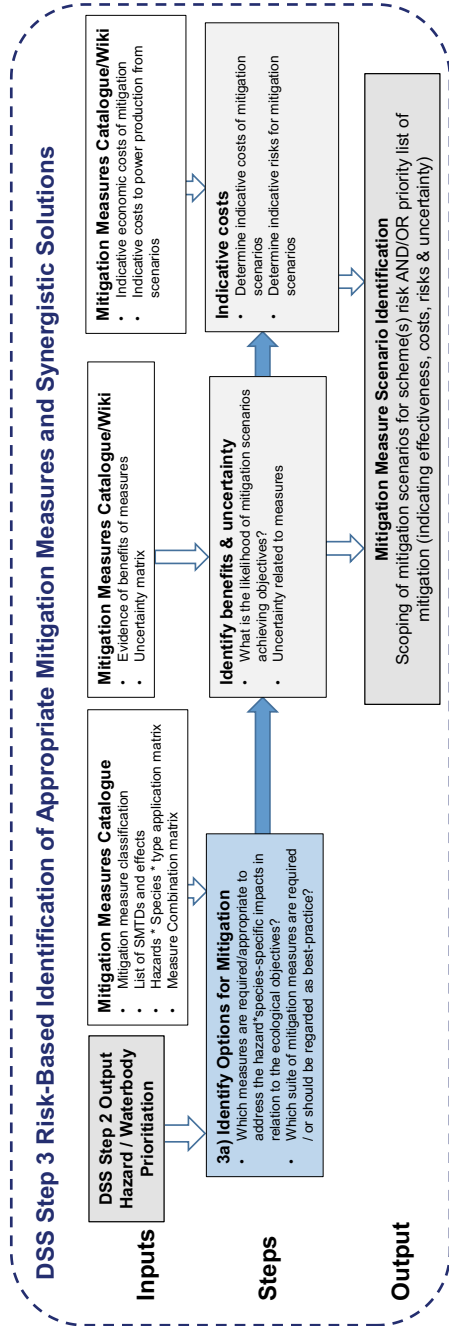


Fig. 15.15 Conceptual flow diagram of Step 3 of the proposed Decision Support System Framework—risk-based identification of appropriate mitigation measures and synergistic solutions. Dark grey = outputs/inputs to/from steps; blue = tasks; light grey = aspects influencing choices; white = FiTHydro results, inputs and matrices

- Direct impacts on fish (e.g. impingement/entrainment and turbine mortality/injury) (Algera et al. 2020; Harrison et al. 2019, 2020).
- Barrier effects (barriers to fish passage and sediment dynamics).
- Habitat alteration/loss effects (impoundment, loss of free flowing river reaches and/or reduced/regulated flows downstream of turbines—including hydropeaking).

In addition to the alteration of flow dynamics caused by the impoundment and abstraction of water for hydropower (particularly in systems where the turbine is located some distance from the barrier, causing a depleted bypass reach) the hydromorphology of the habitats is also influenced by the interruption of sediment transport in the river. These three impact modes in combination with spatial (upstream/downstream) relationship to the barrier/turbine produce a range of spatially explicit hazards that must be addressed in all hydropower risk assessments and mitigation planning. These hazards that are applicable to all hydropower operations are included within the DSS:

- Hydromorphology Upstream—the effects of the impoundment and loss of lotic habitats
- Hydromorphology Downstream—alteration to flows and substrate (as influenced by the effect of the operation on flows)
- Barrier effects—Upstream fish passage and delay
- Barrier effects—Downstream fish passage and delay
- Turbine entrainment and mortality/injury
- Barrier effects—Sediment transport (leading to sediment deposits upstream and sediment deficit downstream of the barrier)

In addition to these hazards, the diverse nature of hydropower types and systems lead to a number of installation specific hazards relevant to specific types of hydropower systems. Based on the summary of the types and systems of hydropower by Harby et al. (2018) three further operation-specific hazards are also considered in the DSS:

- Hydromorphology Downstream—turbine located away from the barrier causing a residual flow (bypass) reach.
- Hydromorphology Downstream—Hydropeaking
- Hydromorphology Upstream—systems operating with a water transfer system from neighbouring catchments creating reduced flow reaches.

All hazard analysis and impact assessments for hydropower therefore need to investigate the (potential) impact of each of these hazards when evaluating proposals for new hydropower or when planning mitigation measures for new or existing schemes. The relative importance of each type of hazard will depend on the spatial context of the scheme, its scale and mode of operation and the fish species present. As such, the DSS provides

screening tools to assess the relative risk of these hazards and to enable the prioritisation of their risk and impact for mitigation and finally the acceptability of a project proposal.

The first step of the FIThydro DSS (Fig. 15.13) characterises the HPP (existing or proposed) in terms of its design (general layout, scale, turbine type, mode of operation), then identify and score the inherent risks posed by the type and size of scheme in relation to its context. There are three tasks in this process: (i) HPP characterises in terms of its location, design and operation, (ii) identifying species/populations at risk, and (iii) identifying and risk assessing key hazards associated with the scheme.

(a) Characterise the HPP—location, operation and features

In this task, the existing scheme or proposal is classified using a structured questionnaire describing the hydropower technology, layout, scale (output, river length, volume/percent of flow used etc.), and mode of operation (run-of-river, hydropeaking etc.). The scheme is also described in terms of its location within a catchment and in relation to other HPPs (when considering mitigation of cumulative impacts).

(b) Identify the Ecological Sensitivity—fish species and populations at risk

This task involves the identification of key sensitive fish species that may be affected by the scheme. This includes consideration of key species that arising from relevant policy directives (e.g. fish species of Habitats Directive annexes). The DSS integrates the species list and sensitivity scoring developed by van Treeck et al. (2017, 2020) and Wolter et al. (2018) (see description of EFHI in Sect. 15.2).

(c) Hazard Analysis and Risk Scoring

A suite of pre-defined features and characteristics of schemes is risk-scored in relation to key hazards posed by hydropower and incorporated into a risk assessment matrix in relation to the sensitivity of affected fishes (most sensitive species identified in Task b). At this stage, an initial risk-based assessment of potential impacts is calculated for the HPP.

The DSS implements hazard scoring templates, comprising scoring systems for eight different hazards relating to the three modes of impacts and in relation to upstream and downstream waterbodies. The hazard scoring templates all have the same basic structure, where answers to a set of questions related to either indicators of the scale of the hazard or factors that pose challenges to effective mitigation, are risk scored based on the potential scale of the hazard. For example, the hazards posed by an HPP is often directly related to the scale of the scheme (e.g. size of an impoundment, height of the barrier, length of a residual flow reach) or its mode of operation (proportion of flow taken, turbine type). As such, quantitative, qualitative and categorical descriptors of key characteristics of HPP

operations, which are known to contribute to the hazards, are used to undertake a hazard specific risk assessment.

The FIThydro DSS tool defines the overall risk as combination of a hazard score (the nature and scale of the hazard) in combination with the sensitivity of the species that are affected. As such, the risk scoring of a hazard is completed within a 2D matrix to determine the risk profile of an HPP and assigns a summary risk score to each relevant hazard. This step underpins an assessment of the hazards posed by a HPP and provides a basis on which to prioritise mitigation measures (per hazard type) and to inform a decision about whether a specific scheme might be acceptable (relative to the location and species affected).

15.4.4 Ecological Status Assessment and Review of Existing Mitigation

The second step in the DSS determines the priorities for mitigating specific hazards posed by the HPP (Fig. 15.14). The first output of this step is a prioritised list of hazards for the scheme(s), with priorities based on actual status of both environmental and energy production objectives for the waterbodies affected. The second output is an evaluation of existing mitigation measures (against permitting/legislative requirements and understanding of site-specific effectiveness) to identify priority hazards for further mitigation and specific measures that may no longer meet best practice and need improving.

(a) Identify Ecological and HPP Policy Drivers

In this task, the HPP is characterised using a pre-defined list of international/national/local policy drivers that are relevant in terms of its spatial location. The decision maker needs to consider which drivers affect the management of the scheme (e.g. WFD, Habitats Directive, European Eel Regulation, other national regulations etc.). This task also identifies high-risk and priority sites or waterbodies based on their policy context. It requires the decision maker to identify specific targets and requirements for mitigation that arise from these directives (and hydropower permitting). Furthermore, this task requires the decision maker to identify and define key national policy objectives and targets that relate to mitigation standards that must be adhered to on a national level. These national legislative and policy requirements differ greatly between different jurisdictions and hence the FIThydro DSS provides scope for the user to freely specify the targets as required.

(b) Identify Ecological Status

Here, the status of the host waterbody for the HPP is determined according to environmental policies that govern the scheme and to evidence for known impacts on fish and waterbodies from national level assessments. The decision maker is directed to select status assessment categories from pre-defined checklists for international directives and input data from national and regional assessments. The aim of this task is to determine the risk

associated with a scheme in terms of causing a net deterioration in ecological/conservation status, and framing the scheme in terms of the objectives and legal requirements for mitigation.

The ecological assessment module in the DSS identifies the ecological objectives and conservation protection for the waterbodies affected by the HPP (or would be affected by a new scheme proposal). This step uses a questionnaire to gather information regarding a range of international environmental legislation and protection (e.g. WFD, Habitats Directive, Eel Regulation etc.) and considers national legislation equivalents to make the tool widely applicable and transferable to other policy contexts. The ecological status assessment acts to prioritise individual hazards based on their legislative drivers for mitigation. The DSS considers the priority based on the level of protection afforded to each waterbody (environmental and conservation objectives) and whether the waterbody meets its objectives.

(c) Identify the effectiveness of existing mitigation measures

When the DSS is used to evaluate the mitigation needs of an existing HPP this prioritisation step is also required to review the performance and efficacy of existing mitigation. This task requires the user to identify specific targets and requirements for mitigation that arise from existing policies (and hydropower permitting). The decision maker has to determine whether there are specific mitigation targets that currently apply (or would apply in the case of re-permitting) to each particular hazard class/mitigation type. National legislative and policy requirements differ greatly between different jurisdictions and hence the DSS allows the user to freely specify targets. These targets may be legislative, permit conditions, best practice principles, minimum standards or locally determined goals. If the user identifies that such targets exist then they are asked to detail them in their own words. This open-ended approach enables the DSS to be truly flexible and transferable between different schemes and regions.

This task enables the decision maker to describe the current mitigation measures in place, which hazards are mitigated and the certainty as to whether they are sufficient/achieving the environmental objects. This is a key task prior to Step 3 in the framework, which aims to identify additional mitigation measures and synergistic solutions addressing unmitigated impacts or failures to meet revised ecological policy objectives.

Firstly, the measures appraisal process involves the definition of measure types in place. The user selects measure types from the predefined FIThydro measures catalogue. It should be noted that these are generic measure classes that do not contain specifications and design of their local installations. Thereafter, the decision maker is required to answer three questions to evaluate the effectiveness of current measures:

Q. Does the implementation meet the current targets/permit requirements for this type of measure in relation to the mitigation of the specific hazards?

Q. What contribution is each measure considered to provide toward mitigating each specific hazard?

Q. Is the suite of mitigation measures currently implemented considered to be effectively mitigating the hazards posed by the HPP?

The first level of evaluation of an existing mitigation measure is whether or not the installed measure meets legislative or permitting requirements or potentially meets current state-of-the-art or best practice for that measure type. Here the user is asked to evaluate the measures per hazard class in respect of specific targets the user has already defined for that hazard. Therefore, this assessment is affected by the nature of the targets used (current or potential future targets) and their level of detail (e.g. qualitative, quantitative). This approach generates flexibility in the system and opportunities to make bespoke evaluations under different scenarios.

The second level of evaluation is in relation to the contribution each measure makes to the mitigation of a specific hazard at the site. This requires the user to make a qualitative evaluation based on local evidence or knowledge gained from similar systems of the importance/effectiveness of that measure for hazard mitigation.

The third level of evaluation requires the user to make a qualitative assessment of the effectiveness of the whole suite of mitigation measures (the current mitigation scenario) on the mitigation of each individual hazard class (recognising that individual measures may contribute to mitigating multiple hazards and that for some schemes individual hazards are addressed with multiple measures). To support this, the user is presented with the outputs of the ecological status assessments in Step 2 to prompt the user to consider the mitigation effectiveness with respect to the actual ecological and conservation status of the affected waterbody.

Whilst the status of the waterbody acts as a driver for further mitigation (if environmental/conservation objectives are not met) the evaluation of the actual effectiveness of the mitigation scenario for a specific hazard could be determined from a range of different evidence (and levels of (un)certainty). This includes local empirical evidence or evidence obtained from similar schemes. It is anticipated that the user would select a definitive answer (high levels of confidence) where local empirical evidence of effectiveness has been obtained. Otherwise, all other answers relate to less detailed, or inconclusive, assessment of sufficiency of the mitigation scenario. The answer to these questions determines whether further or updated/improved measures are required for specific hazards.

The overall output from this step is a risk-based prioritised list of scheme-specific hazards/measures that need further mitigation or improvement to meet objectives/permit requirements/recognised best practice (Fig. 15.16). At this stage in the DSS, the assessment outputs are in the form of three matrices that characterise the scheme based on: (1) their inherent risk to affected fish populations; (2) the environmental/conservation

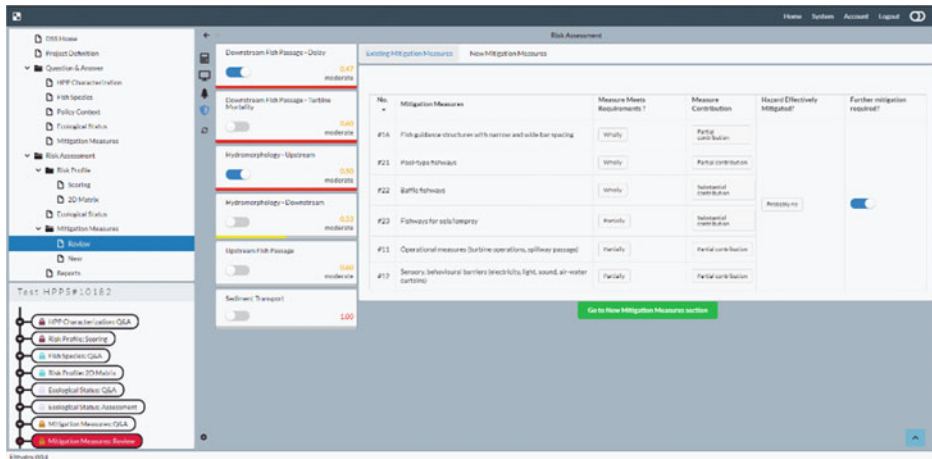


Fig. 15.16 FIThydro DSS existing mitigation measure evaluation summary screen reflecting the answers given to the evaluation questions and indicating for each hazard whether further mitigation is considered to be required

objectives and current ecological status; and (3) for existing schemes the effectiveness of current mitigation. These assessments are used to rank hazards as priorities for mitigation when the user is required to identify potential mitigation options (Sect. 15.4.5).

15.4.5 Identification of Appropriate Mitigation Measures and Synergistic Solutions

Whilst the types of hazards posed by HPP are similar across all schemes, hydropower is a very site-specific technology (Harby et al. 2018). As such, design and operation of the scheme are determined by the specific characteristics of the hydropower opportunity being exploited. Therefore, impacts realised from key hazards vary between schemes, as does the potential and methods required for impact mitigation. It follows that the ability to identify site- and scheme-specific effective mitigation measures is key in project evaluation and planning.

However, identification and implementation of appropriate and effective mitigation measures is also one of the major challenges in hydropower planning and mitigation. Whilst some countries have mitigation measures for upstream fish passage and modified flow conditions detailed in legislation, prescribed standards for mitigation measures are often lacking and have to be identified on a case-by-case basis (Kampa et al. 2017). Decision-making is, therefore, often limited by the availability of evidence and knowledge required to underpin the planning of new or suitable mitigation measures. This is a major

challenge that needs to be addressed to inform decision making in hydropower planning and permitting that needs improved knowledge and decision support.

Thus, the third step of the decision framework supports identification of most appropriate mitigation measures and potential synergistic solutions for high-risk hazards and impacts that have been determined in the preceding steps (Fig. 15.15). The hazard-species-risk-matrix that results from Step 1 of the framework is cross-referenced to the review of existing mitigation measures and ecological status prioritisation matrix arising from Step 2 of the framework and directs decision maker to identify what alternative mitigation options would be appropriate and potentially most cost-effective. This step also aims to scope different options based on an assessment of benefits of mitigation measures and the likelihood of their success in achieving environmental objectives (risk and uncertainty). This step also provides the decision maker with summary information of indicative costs of mitigation measures selected both in economic terms and loss of power production.

In this task, the decision maker identifies options for:

- New mitigation measures
- Existing measures that need to be improved (e.g. modified designs based on improved knowledge and new developments)
- Measures/hazards that will need further investigation within further detailed project planning.

During the selection process, the user is presented with the FITHydro mitigation measures catalogue and asked to identify which measures they would like to select or modify. The user is asked to do this per hazard class and is directed to approach this in sequence from the highest priority hazard (highest prioritisation score in relation to ecological status) to the lowest. The measure selection process is aided by the DSS automatically:

- Filtering the available measures catalogue based on characteristics of the scheme (filtered using definitions in the catalogue and questions answered by the user at the start of the project)—thus only measures that are applicable at the site are available to the user.
- Filtering the measures catalogue based on a mapping of measures to different hazard classes—thus within each hazard, only measures known to be effective at mitigating that hazard are available for selection.

To enable the user to make informed decisions and selections the DSS also provides the user with an overview from the measures catalogue of:

- The compatibility of the measure—a comparison of all potential measures, summarising the known compatibility of particular measure types with those measures previously selected/installed. This could direct the user towards selecting measures

that will maximise the effectiveness of mitigation and not select measures that may conflict with solutions already chosen/in place.

- A summary of known effectiveness of measure (scored in Harby et al. 2019) in relation to mitigation of:
 - Instream Degradation
 - Shoreline and Off-Channel Degradation
 - Barriers Obstructing Upstream Migration
 - Barriers Obstructing Downstream Migration
 - Sediment Transport
 - Surplus of sediment upstream
 - Deficit of sediment downstream
- Level of certainty in the effectiveness
- Technological readiness level (TRL)
- Whether measure potentially results in loss of power production
- Measure's requirement for maintenance
- Indicative cost of measure

In addition to these summary metrics, the user is able, for each measure type, to access the FITHydro wiki page with a full measure description and supporting evidence for its effectiveness.

The key output to this step is a high-level scoping report that identifies hazards and risks and contains an options appraisal of potential mitigation measures types appropriate to the scheme and likely to meet objectives.

15.4.6 Summary and Scope of the FITHydro DSS

The FITHydro DSS open access web tool (<https://fithydro.eu/dss>) is aimed at regulators, consultants, developers and operators and is orientated around a single, risk-based, planning structure. The planning/decision framework within the DSS tool is implemented screening and scoping tool for the high-level assessment of HPP mitigation. The DSS summary assessment of the risks posed by a scheme focuses on prioritising hazards for mitigation in relation to ecological status and objectives and then selecting appropriate hazard specific mitigation measures (filtered from a catalogue of potential measures).

The DSS was designed to directly use outputs and innovations from the research in FITHydro or to act as a gateway to more specialised tools and results applicable to their specific circumstances. Throughout the decision-making process, the DSS aims to enable decision makers to make informed and evidence-based choices. To achieve this the DSS tool integrates with the FITHydro wiki (<https://www.fithydro.wiki/>), a resource that contains descriptions of impacts and measures, and evidence and examples from case studies

used within the FITHydro project. Wiki links within the DSS to enable users to access supporting materials detailing the summary data held by DSS and to find detailed descriptions of solutions, methods, tools and devices that could be applicable for further site-specific impact assessment and mitigation.

The tool is intended to support existing decision protocols and harmonise approaches by providing procedures for both the initial appraisal and screening and defining options and criteria for best practice in mitigation; thereby supporting the implementation of existing regulations. However, like all DSS the FITHydro system does not provide the decision maker with the answer but directs the user to identify the knowledge, evidence and means to reach a conclusion and understand the options available for improved mitigation of hydropower schemes to promote sustainable fish populations.

Acknowledgements The work of Nguyen Ngoc Dzung of Technical University Munich, who developed the web interface for the DSS, deserves special mention at this point. Without his contribution, the DSS could not have been developed and realized in the way it was in the end.

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